National Environmental Research Program

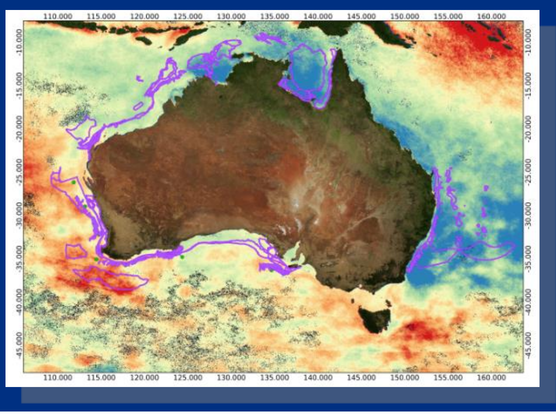
MARINE BIODIVERSITY hub

A hierarchical risk assessment framework for ecosystem based management

Piers Dunstan, Jeff Dambacher, Nic Bax, Tony Smith, Beth Fulton, Paul Hedge, Alistair Hobday, Scott Foster.

Theme 2: Supporting management of marine biodiversity

1 September 2015



Australian Government Department of the Environment · Museum Victoria · Charles Darwin University · UTAS · CSIRO · Australian Government Geoscience Australia · The University of Western Australia · Australian Government Australian Institute of Marine Science

www.nerpmarine.edu.au

Enquiries should be addressed to:
Piers.Dunstan@csiro.au

Copyright and Disclaimer

Acknowledgement

This work was undertaken for the Marine Biodiversity Hub, a collaborative partnership supported through funding from the Australian Government's National Environmental Research Program (NERP). NERP Marine Biodiversity Hub partners include the Institute for Marine and Antarctic Studies, University of Tasmania; CSIRO Wealth from Oceans National Flagship, Geoscience Australia, Australian Institute of Marine Science, Museum Victoria, Charles Darwin University and the University of Western Australia.

Important Disclaimer

National Environmental Research Program

MARINE BIODIVERSITY hub

Contents

List of Figures

List of Tables

1. SUMMARY

There is general agreement that assessment of risk in the marine environment needs to move toward an ecosystem approach to account for the single and cumulative impacts from multiple sectors that operate within the world's oceans.

Despite there being fewer marine activities than in terrestrial environments, marine systems are challenging to manage on an ecosystem basis as a result of their complexity, high degree of connectivity and difficulties associated with observing ocean processes, flora and fauna. These challenges can make it difficult for researchers to know how to make best-use of available scientific information to inform policy makers about options for ecosystem management. A broad range of scientific tools and approaches have emerged to attempt to meet these differing needs and together these challenges and choices have stymied decision makers.

There is a clear need to develop a process that can assist governments and other decision makers to reduce the uncertainty around the risks of significant impacts in ecosystem based management. An important consideration in developing a framework for risk-based approach to ecosystem management is clarifying the terminology associated with the assessment, this is particularly important for facilitating collaboration between researchers and policy makers.

We suggest that a productive way to approach this would be to use a hierarchical approach where a range of tools can be used to identify activities that have a high risk of significant impact. We use values (eg conservation, resource or community) that have been described through an expert process to identify the relevant subsystem for management. The first level builds a conceptual model of the relevant subsystem and identifies the pressures that act on that subsystem. The second level uses mathematical qualitative models to refine the understanding of the system and to reduce the uncertainty around the system structure. The final level uses quantitative and qualitative models to identify specific thresholds, management trigger points and scenarios. Each level reduces the uncertainty in decisions but increase the costs and time taken.

The hierarchical framework proposed in this paper provides scientists and policy makers with guidance and a common lexicon for assessing cumulative risks and estimating impacts to marine ecosystems. The framework provides for a cost-effective and consistent approach to assessments, accommodating a broad range of marine environment assessment cases, leading to priorities for action. The approach acknowledges the importance of ecosystem models for estimating cumulative risks and provides a frame for understanding how they can be cost-effectively and consistently applied to estimate impacts and improve understanding.

2. INTRODUCTION

There is general agreement that assessment of risk in the marine environment needs to move toward an ecosystem approach to account for the single and cumulative impacts from multiple sectors that operate within the world's oceans. It is clear that the number and breadth of human activities in oceans and coastal zones are increasing (Halpern et al. 2008, Halpern et al. 2009, Coll et al. 2012). There is a long history of assessing risk of impacts from single sources of disturbance and pressure (eg fisheries, shipping, and pollution) but understanding how to cost-effectively assess risks at ecosystem scales and across multiple sectors remains a substantial challenge that is shared by both scientists and policy makers.

An ecosystem approach would need to confront the difficult convergence of conserving or managing multiple species/groups with multiple activities potentially impacting the ecosystem. The ecosystem approach has been variously described as including ecosystem based management, marine spatial planning and sector specific assessments – most notably ecosystem based fisheries management and the ecosystem approach to fisheries. A constructive step for determining a cost-effective ecosystem approach is establishing a framework to guide scientists and policy makers through a process to assess potential risks and efficiently getting to the point where priorities for action emerge (eg to avoid, mitigate or offset). At this point unacceptable risks are identified that require further investigation through more targeted and detailed ecological risk assessments. For the framework to have broad utility it needs to be specific enough to consistently guide the user through the issues and options that need to be considered to undertake a cost-effective assessment, but general enough that it can be applied to and accommodate the majority of cases or ecosystems.

Despite there being fewer marine activities than in terrestrial environments, marine systems are challenging to manage on an ecosystem basis as a result of their complexity, high degree of connectivity and difficulties associated with observing ocean processes, flora and fauna. These challenges can make it difficult for researchers to know how to make best-use of available scientific information to inform policy makers about options for ecosystem management. A broad range of scientific tools and approaches have emerged to attempt to meet these differing needs and together these challenges and choices have stymied decision makers. One approach that shows promise is to first assess the major impacts on a particular asset or system, identifying the relative contributions of each impact on observed status, the synergies between impacts (ie the cumulative impact) and then identifying the leverage points where achievable intervention can have a larger effect (*sensu* Bax et al 1999). This is an intentional step away from proposed all inclusive integrated management proposals. While attractive in theory such approaches have failed to get traction operationally. Consequently the form of pragmatic ecosystem management outlined here does not imply managing the entire complex system, rather managing individual activities across specified subsystems, while recognising the broader implications of these activities.

An important consideration in developing a framework for risk-based approach to ecosystem management is clarifying the terminology associated with the assessment, this is particularly important for facilitating collaboration between researchers and policy makers. Risk and impact are variously defined and measured, and even within relatively restricted single sector domains there is no single agreed approach – taking fisheries as an example compare the discussions in Hilborn et al. (2001), Fletcher (2005), Astles et al. (2006), Scandol et al. (2009) Sharp et al. (2009) Hobday et al. (2011) and Sethi (2011). The different definitions and measures of risk sometimes reflect different lexicons, but more often reflect fundamental differences in method driven by facets such as the objectives and scope of the study and the availability of data. Moreover, pressure, impact, risk and threat are often confused so that it can be difficult to distinguish between these from a scientific or management perspective. Here, we use a clearly defined hierarchy of descriptions of human activity. Pressures are the actual activities that occur in the ocean. These are varied, from large scale (eg climate change) to small scale (ie sewerage outfalls and boat ramps). Impact is the change in the state of the variable or ecosystem value of interest as a result of the pressure and where there is more than one pressure impacting the cumulative impact can be calculated. Risk is defined as the likelihood of a significant impact on a value of concern and is largely synonymous with concept of a threat.

There are a number of approaches to estimating cumulative risk and impact that deal with uncertainty and ecosystem function. The most common approach to calculating cumulative impacts to date has been through scoring methods. Typically, the mapped pressures are standardised (Halpern et al. 2009) and aggregated based on either the sum or weighted sum of the pressures. The weights can be calculated based on expert opinion or some form of prior analysis (eg Coll et al. 2012). When based on expert opinion there are a number of different approaches that can be applied that provide differing levels of rigour. However, processes that use expert opinion can contain bias in the level of risk placed on different outcomes (eg Garthwaite and O'Hagan 2000, Garthwaite et al. 2005, Kadane And Wolfson 1998, Kynn 2008) and can be difficult to update because the expert process used to determine the risk is not repeatable, or at least will confound the effects of the new information with changing composition of experts, changing opinions of continuing experts and other related developments. Updating expert derived risk is particularly problematic when cumulative impact is used within an EBM framework that includes a monitoring scheme created to track changes in the ecosystem. Expert derived risk is based on existing information and so by definition improved understanding of the structure and function of the observed system will lead to an improved understanding of the pathways through which each pressure acts and the relationship between pressure and impact which will lead to experts potentially shifting understanding in addition to the direct effect of the new information itself. Consequently a transparent and consistent means of estimating risk is required for this adaptive approach.

Moving from expert based judgement to quantitative assessment requires a rapid rise in complexity and associated data needs. More quantitative approaches rely on access to appropriate data and can take the form of statistical (eg Foster et al. 2014) or numerical simulations (eg ecosystem models, Fulton et al. 2011), or borrow strengths from both expert and mathematical approaches (eg qualitative models, Dambacher et al. 2009, 2010). The challenge is to understand how and when all these different ideas and approaches can be used within an assessment framework. In moving towards improved understanding there remains the need to maintain flexibility and an adaptive approach while retaining realistic expectations on what science can deliver and what resources are available.

An effective, flexible and resource appropriate approach to risk assessment can be found in Ecological Risk Assessment for the Effects of Fishing (ERAEF, Hobday et al. (2011)) and the FAO Ecosystem Approach to Fisheries (Fletcher and Bianchi 2014). The ERAEF approach was developed to provide a framework and set of tools to assess the ecological impacts of fishing on species (including target, bycatch and protected species), habitats and communities, and was an important step in a move within Australia toward Ecosystem Based Fisheries Management (EBFM, Smith et al. 2007, Smith et al. 2008, Smith et al. 2014). It was incorporated into an adaptive management approach by the Australian Fisheries Management Authority, which they termed ecological risk management. The ERAEF approach allows for three hierarchical levels of increasingly quantitative analysis, with associated increases in data requirements and analytical costs. At each level, a risk assessment occurs that determines the level of risk of activities to assets, and activities with risk below a set threshold need not be assessed further. This approach has been successfully used in managing non-target fisheries management issues for all Australian federally-managed fisheries and some state fisheries, and a modified version was adopted by the Marine Stewardship Council (https://www.msc.org/). ERAEF tools to level 2 have also been adopted for habitat impacts (Williams et al, 2011), but level 3 analyses are generally required to assess trophic and ecosystem-level impacts.

In this paper we focus on the technical challenge of developing a framework for assessing cumulative risks to estimate the risk of impact to ecosystem components. We describe a hierarchical approach, similar to ERAEF, which can inform decision-making for managing risk and improving ecosystem understanding. The approach recognises that while ecosystem modelling tools are an important part of estimating impact there are a broad selection of these tools and that each of these have strengths and weaknesses depending on the context in which they are applied. Importantly the framework provides researchers and policy makers with guidance for cost-effective and consistent approaches to estimating cumulative risk to marine ecosystems.

3. RATIONALE FOR A HIERARCHICAL APPROACH

There are three key concerns that need to be addressed: (1) there are multiple pressures on the marine environment; (2) some or all of these will have the risk of impact on ecosystem values; and (3) it is uncertain which ones will have a high risk of impact, what the magnitude of that impact will be and what are the likely synergistic effects. We need a framework that allows rapid assessment and elimination of low risk pressures and a graduated response as risk increases, thus focusing assessment (and management) effort either where risks are greatest and/or where intervention can have most affect. Issues of knowledge, data availability, cost, and uncertainty all limit the application of many tools and approaches. It might be desirable to have a single tool that could always be used to decide on the optimal/most efficient management option, but the number of circumstances where this is possible is small, and policy makers often prefer a set of options that they can test against additional non-scientific criteria. A hierarchy of tools, moving from simple, rapid and low cost tools to progressively more complex and costly tools would support the prioritisation that managers will typically need to undertake.

Fundamentally, the successful application of this approach is the identification of the **relevant subsystem**, based on desired **values** (eg conservation, sustainability, resource). The goal is not to try to account for all species and processes in an ecosystem, but rather to identify a relevant subsystem that captures the essential dynamics supporting the desired values of the system and is tailored to the management objectives (Dambacher et al. 2009, 2015). In our example the objective is sustainable management and the relevant subsystem is the ecological system encompassing the asset or value of concern, including the pressures that directly or indirectly affect them. This is similar to the "abstraction of ecosystems into sub-systems thought to be most influential to the management issues at hand" (Levin et al. 2009) that underlies the NOAA approach to EBFM.

The assessment hierarchy we propose has three levels with a preliminary scoping step to identify values (Figure 1). The first level is an expert based assessment of the interaction between the values in the relevant subsystem and identified pressures. This first level of assessment is based on a general conceptual model of the system, while assessment levels two and three require an increased use of mathematical models that provide greater understanding, prediction and scope for management interventions (Levins 1966). The second level employs qualitative mathematical models that use the information from the first level to build a more robust understanding of the relevant subsystem. The third level combines the use of qualitative and quantitative models that require extensive data and resources. Each of the previous levels provides the context and justification for further investigation of risk to ecosystems/values/assets (ie triggers for progression to the next level in the hierarchy). While the three levels of assessment are laid out as a three-stage progression in Figure 1, they are, in practice, intended to provide a progressive feedback between modelling, monitoring and management activities (Levins 1966).

4. LEVELS OF THE CUMULATIVE RISK ASSESSMENT HIERARCHY

4.1 Enabling conditions

An essential task of any EBM framework is the definition of scope: who are the stakeholders, what domain that is being considered and what are the active governance structures (Figure 1)? The level of inclusiveness that is adopted here can influence how the process moves forward and determine the objectives of the assessment. The step should include the identification of species and/or areas of particular importance or focus for the assessment.

There is considerable experience in identifying the relevant species (eg ERAEF), but identifying areas of interest is less well developed. A pragmatic approach is to identify areas that contain the well identified, ecologically coherent systems that contain features that could be both responsive to management (and impacted by activities) and perform an ecologically or biologically important function. This is also an acknowledgement that there are significant areas of the ocean that we do not have sufficient scientific information to manage based on evidence. We are choosing to focus on the areas where there is sufficient information to articulate the values for that area (ie to at least level 1 in our hierarchy). There are a number of ways that these areas can be identified – eg Key Ecological Features (KEF; Dambacher et al. 2012), Biologically Important Areas (BIA), Ecologically or Biologically Significant Marine Areas (EBSA; CBD (2008)), and other approaches listed in table 1. However the unifying attribute is the identification of ecological features that are valued for their productivity or biological diversity and a subset of the ecosystem within that area that supports those ecological features within a spatially defined area. For KEFs, these elements are described as the **relevant subsystem**, which is a description that applies equally well to EBSA, KBA and all the other area-based descriptions, as well as attributes described as *values*.

The information sources that can be used to identify productivity and biodiversity values are diverse and will depend on the regional, national and local capacity. An important component of understanding the values will frequently be the knowledge held as traditional/local knowledge by communities and the processes described here could equally be applied to community-level management efforts (eg Community Based Fisheries Management; CBFM, SPC 2010). In areas with more scientific capacity, existing and future research surveys will provide significant sources of information for the identification of biodiversity values. As this is an adaptive approach, identifying the biodiversity values to be considered should be based on best available scientific information.

Table 1: Examples of Biodiversity Values

Name	Description	URL
Key Ecological Features & Biologically Important Areas	Key ecological features (KEFs) are elements of the Commonwealth marine environment in the marine regions that, based on current scientific understanding, are considered to be of regional importance for either the region's biodiversity or ecosystem function and integrity. Biologically important areas are areas where a protected species displays a biologically important behaviour such as breeding, foraging, resting and migration	http://www.environment.gov.au/marine/marine-bioregional-plans
Ecologically or Biologically Significant Areas	Scientific criteria for identifying ecologically or biologically significant marine areas in need of protection in open-ocean waters and deep-sea habitats.	http://www.cbd.int/ebsa/
Canadian EBSA	An area that has particularly high Ecological or Biological Significance, to facilitate provision of a greater-than-usual degree of risk aversion in management of activities in such areas.	http://www.dfo-mpo.gc.ca/csas/Csas/status/2004/ESR2004_006_E.pdf
Victorian Marine Asset	Significant marine environmental assets which have been identified on the basis of their environmental value (at statewide, bioregional or local significance scale) for marine biodiversity and/or marine ecological processes	http://services.land.vic.gov.au/SpatialDatamart/dataSearchViewMetadata.html?anzlicId=ANZVI0803004772&extractionProviderId=1
Environmental values in Norwegian marine areas	A marine area has high environmental value when it is important for preserving the diversity, productivity and special functions of the ecosystem, such as spawning or nesting areas.	http://havmiljo.no/, Ottersen et al. 2011

Name	Description	URL
Key Biodiversity Areas	Key biodiversity areas are places of international importance for the conservation of biodiversity through protected areas and other governance mechanisms. They are identified nationally using simple, standard criteria, based on their importance in maintaining species populations.	https://www.iucn.org/about/union/secretariat/offices/iucnmed/iucn_med_programme/species/key_biodiversity_areas/
IBA	The 12,000 IBAs represent the largest global network of important sites for biodiversity. They are identified using internationally agreed criteria applied by local experts	http://www.birdlife.org/worldwide/programmes/important-bird-and-biodiversity-areas-ibas
PSSA	A Particularly Sensitive Sea Area (PSSA) is an area that needs special protection through action by IMO because of its significance for recognized ecological or socio-economic or scientific reasons and which may be vulnerable to damage by international maritime activities.	http://www.imo.org/OurWork/Environment/PSSAs/Pages/Default.aspx

4.2 Level 1: Identification of hypotheses about interactions between pressures and values

4.2.1 Description of Process

Once a process of identifying the spatially bounded values has been completed, a process of identifying and describing the pressures that may act on the ecosystem values should be undertaken (Figure 1). This allows for the articulation of perceived interactions between the pressures and values, enabling the description of conceptual models of the relevant subsystem. Conceptual models play an important role in organising understanding and communicating the links between different components in the system. They formalise what may otherwise remain in an individual expert's head and provide a shared level of understanding by all parties.

The level one approach identifies insights and hypotheses about where and when pressures are affecting the relevant subsystem. For example in the document supporting Australia's marine bioregional plans:

> *"If a conservation value is, or is likely to be, affected detrimentally by multiple pressures, and at least one of the pressures has been assessed as of concern, it is considered to be a regional priority. Other key considerations in determining pressure-based regional priorities included issues of scale, legislative responsibility, conservation status, effectiveness of existing management arrangements, and level of uncertainty about distribution, abundance and status of conservation values and the pressures acting on them."*
> *(DSEWPaC 2012)*

4.2.2 Tools used

There are a number of tools and approaches, remembering that the expectation around a level 1 analysis is that it is a simple and rapid filtering of risks, it does not need to be particularly quantitative or quantitatively complex. The simplest means of analysis is the direct examination of the interaction of the ecosystem values identified in the relevant subsystem and the pressures thought to interact with that subsystem, allowing a conceptual model of the relevant subsystem to be constructed. There are two key components to this. First, the pressures that occur within the area need to be identified and assessed to see if there is possible interaction between the pressures and the area identified within the relevant subsystem. If there is no possible spatial overlap and if the pressures could not reasonably be expected to interact with the values of interest then the pressure should be considered a low risk with no further consideration required. Second, expert elicitation can be used to identify and rank the potential risk of impact from pressures on the values in each relevant subsystem. The elicitation can be either structured or unstructured. Structured elicitation is preferred (as it confers some degree of consistency), but it is not always possible and so unstructured elicitation should not be ruled out if alternatives are not available.

Unstructured elicitation may involve a consensus process where a group of experts identify the potential interactions between pressures and values on a scale of consequence (eg pressures are "of concern", "of potential concern", "of less concern", "not of concern", "data deficient or not assessed").This type of approach has been used in many fora (eg Marine Bioregional Plans (DSEWPaC 2012), Community Ecosystem Approach to Fisheries Management (SPC 2010)). While this provides a quick simple answer it does not allow for ranking the pressures and it limits the ability to compare between different areas. It also makes it difficult to prioritise in a consistent manner, particularly across different relevant subsystems. In contrast, a structured process of expert elicitation allows for the relative ranking of the interactions between pressures and values (eg Garthwaite and O'Hagan 2000, Garthwaite et al. 2005, Kadane and Wolfson 1998, Kynn 2008). It also allows for the scoring of the interactions relative to each other and provides a quantitative estimate of the experts' understanding of the relative impacts on the values identified in the areas of interest. A relative ranking will identify the risk of different pressures relative to the pressures within the same relevant subsystem.

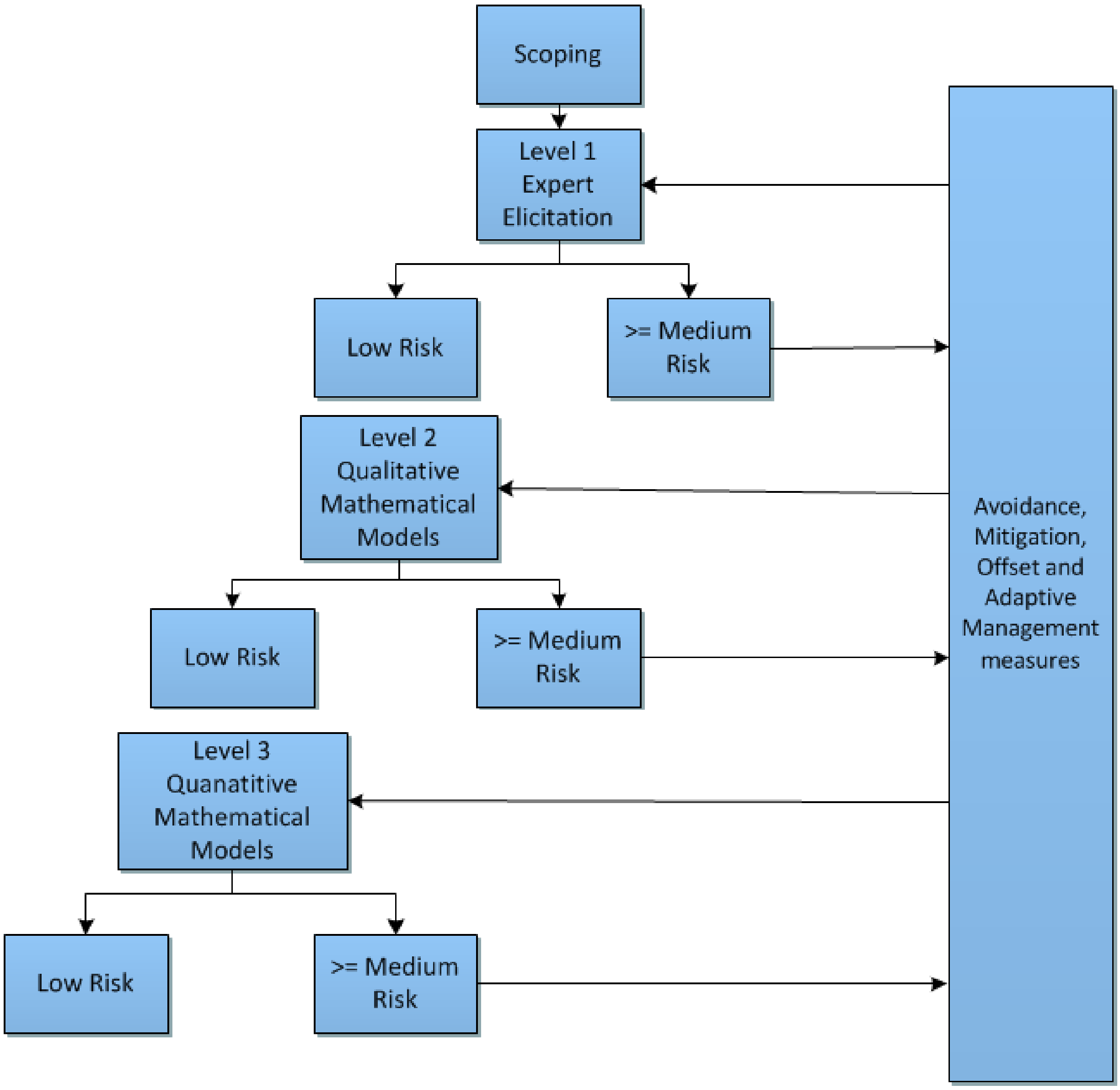


Figure 1: Proposed framework for hierarchical ecosystem risk assessment

4.2.3 Transition to level 2

Before transitioning to a higher level of analysis it may be appropriate to consider whether sufficient information is already available to manage the pressures and monitor the success of this management. If all the risks are identified as low then progression may not be necessary. Alternatively, it may be decided that there is no acceptable level of risk for values identified in the relevant subsystem and the pressure would be managed to remove its impact over part of or all of the relevant subsystem, in which case progression is again unnecessary as a decision can already be made (Figure 1).

If the pressure cannot be removed from all or part of the relevant subsystem and the assessment has identified the pressure is a concern (ie greater than a predefined threshold) then there are two options. Either the pressures of concern can be managed based on the information made available through level 1 (ie avoidance, mitigation, offset and adaptive management measures) or it might be appropriate to transition to a higher level of analysis (ie level 2) that would increase the understanding of the risk posed on the relevant subsystem and improve the identification of the scale or type of management intervention that could be used to minimise or remove the pressure at an acceptable cost to society. However, this desire for more information must be weighed against the cost of the increased information requirements and increased duration to complete assessments at higher levels.

Table 2: Examples of Level 1

Example of Level 1	Description	Reference
Marine Bioregional Plans	Marine bioregional plans have been developed for four of Australia's marine regions - South-west, North-west, North and Temperate East. Marine Bioregional Plans will help improve the way decisions are made under the Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act), particularly in relation to the protection of marine biodiversity and the sustainable use of our oceans and their resources by our marine-based industries	http://www.environment.gov.au/marine/marine-bioregional-plans
Assessment of anthropogenic threats to marine protected areas in Victoria	A preliminary threat assessment approach for marine assets in Victoria was developed in 2011-12by the then Department of Sustainability and Environment (DSE) , in collaboration with the CSIRO and the then Department of Primary Industries (DPI), based on Australian Standard risk assessment guidelines. This approach is hierarchical, and builds on the ecological risk assessment approach that has been successfully applied to of Australia’s federally managed fisheries	http://www.veac.vic.gov.au/documents/Jenkins%202013%20Threat%20assessment%20method.pdf
AFMA ERAEF Level 1	The risk assessments are applied hierarchically and are an efficient means of screening out low-risk activities and focusing increasing attention on those activities assessed as having a greater environmental impact on Australia’s fisheries.	http://www.afma.gov.au/managing-our-fisheries/environment-and-sustainability/ecological-risk-management/
FAO EAF Level 1	Identify all relevant assets and issues for the fishery across each of the EAF components (ecological wellbeing, human wellbeing and ability to achieve)	http://www.fao.org/fishery/eaf-net/topic/166253

4.3 Level 2: Qualitative mathematical models of ecosystem impact and risk

4.3.1 Description of Process

A more complex understanding of the dynamics and structure of the relevant subsystem within an ecosystem and its components can be developed using qualitative mathematical models (Figure 1). With increased understanding of the biodiversity values and ecosystem components, it is possible to construct ecosystem models that allow for a more informed, albeit qualitative, estimate of the cumulative impacts of pressures on ecosystem values (eg Dambacher et al. 2009, Dambacher et al. 2010, Hosack and Dambacher 2012, Anthony et al 2013). A semi-quantitative process is also used within level 2 analysis of ERAEF (Hobday et al. 2011). The ERAEF uses a semi-quantitative productivity susceptibility analysis (PSA), scoring fisheries on the productivity of species and the susceptibility of each species to the types of fisheries gear used. Both approaches take elements of the information gathered as part of level 1 and incorporate them into a more quantitative information rich framework.

4.3.2 Tools used

Qualitative mathematical models identify the nodes/ecosystem variables within the relevant subsystem and how those nodes are connected to each other (Levins 1974, Puccia and Levins 1985). This approach uses much of the information collected during scoping and level 1, but applies additional analysis to improve understanding of the relevant subsystem. It requires an understanding of how the values identified in the area linked to each other and to other nodes in the ecosystem (ie how the component parts of the relevant subsystem relate to each other) and the points where pressures interact with either the values or other ecosystem components. The links between the nodes in the qualitative model are given a score of + (positive effect), 0 (no interaction) or - (negative effect), but there is no attempt to quantify the strength of these interactions. This gives these models significant flexibility and allows multiple representations of the structure of the relevant subsystem to be created and compared, but does not provide information on the strength of the interactions. Thus, the influence of one node on another, irrespective of how many other links and nodes exist between them can only be qualitatively specified+, 0, - or unknown (indicated by visual representations of the model) Importantly, these models can capture indirect effects and causal chains (eg trophic cascades are a well-known example where suppression of a predator at a high level can also reduce prey two levels below because the abundance of the intermediate consumer increases)

The strength of the qualitative modelling approach is that the models can formally capture information about the structure of the relevant subsystem, particularly for components and that are difficult to measure, and can draw information from knowledge bases that are hard to access quantitatively (eg social or cultural knowledge). Because they are derived from our current understanding of the system they can be developed, analysed and updated rapidly as new information becomes available. The dynamics of the system can be understood and predicted through examining the system's qualitative structure and feedback properties. In this way the level two approach provides qualitative predictions about how cumulative risk and impact are likely to affect the specific components of relevant subsystems and which components would need to be monitored to unambiguously detect and attribute impacts to the different pressures.

4.3.3 Transition to level 3

The need to transition from level 2 to 3 can be assessed based on similar conditions to the transition from 1 to 2. If pressures can be removed or managed based on information obtained at level 2 then progression to level 3 is unnecessary. If the pressure cannot be removed, reduced or restricted from the relevant subsystem and the assessment identifies pressures that are of concern (ie cause negative or uncertain outcomes for the system values of interest) then there are two options. Either the pressures of concern can be managed based on the information made available through level 2 or a transition to a higher level of analysis (ie level 3) may be appropriate, as that would increase the understanding of the risk posed on the relevant subsystem. This decision must be made with the clear understanding that a transition to level 3, a fully quantitative analysis, implies significantly more expense and complexity.

Table 3: Examples of Level 2

Examples of Level 2	Description	Reference
Ecological Indicators for Australia's Exclusive Economic Zone	Ecological indicators reduce the complexity of real-world systems to a small set of key characteristics that are useful for management and communication purposes.	Hayes et al. 2012
ERAEF PSA Analysis	*Ecological risk assessment for the effects of fishing*,	Hobday et al. 2011
Northern Prawn Fishery	Qualitative mathematical models to support ecosystem-based management of Australia's Northern Prawn Fishery	Dambacher et al. 2015
Qualitative modelling in the Great Barrier Reef	A Framework for Understanding Cumulative Impacts, Supporting Environmental Decisions and Informing Resilience-Based Management of the Great Barrier Reef World Heritage Area	Anthony et al. 2013

4.4 Level 3: Quantitative analysis of Ecosystem Impact and Risk.

4.4.1 Description of Process

In some of situations a more quantitative understanding of the risk of different pressures will be needed to decide on thresholds and trigger points for actions, or provide managers with an increased knowledge of how to choose between potential future scenarios. This will be particularly relevant when previous levels have indicated that activities may be high risk and there is insufficient information to mitigate pressures as a result of the assessments at previous levels (Figure 1). This is the only level where a fully quantitative analysis is undertaken and where information from all levels should be integrated and used.

There are a significant number of analytical options that exist to address ecosystem level analyses and the challenge is choosing the approach that meets the objectives of the assessment and the time and budget constraints. The first constraint for this approach is the availability of data or the ability to obtain additional data. Numerical data is expensive both in terms of cost and time to analyse and if there is existing data that can be used to address the objectives of the assessment then it is possible to shorten this aspect of the process. Alternatively, additional information may be obtained through a monitoring program (Hayes et al. 2015) or scientific surveys that explore the response of the system with adaptive management. The implementation of a monitoring program would be a reasonable response to the absence of data, using information obtained in levels 1 and 2. The program would need to clearly identify how the additional information would be used to update management options, appropriate trigger points and a process for updating the analysis of the monitoring data (Hayes et al. 2015).

4.4.2 Tools used

Here we are guided by a strategy of model building that recognizes a practical trade-off between realism, generality and precision when building and analysing models of complex systems (Levins 1966, 1998). To obtain a manageable and useful model, one typically sacrifices one attribute for the other two. Qualitative process models emphasize generality and realism, but lack precision, while quantitative process models can be both precise and realistic but are not generalisable (ie application of model to changed circumstance requires re-parameterization). A third approach is through statistical models, which emphasize precision and generality. Here there are precise insights into the general pattern of correlations among variables, but at the cost of causal understanding of the processes involved.

In practice, a robust strategy considers all three modelling approaches, such that models are mutually informative and build upon the strengths and insights that each approach provides.

These models contain only the sign (+, –, 0) of species interactions, and not their precise magnitude or strength. In this approach, one sacrifices precision in a model for generality and realism (Levins 1966). Qualitative mathematical models can incorporate components and processes of an ecosystem that are important yet difficult to measure, and can be constructed and analysed relatively rapidly, thus allowing for comparison of alternative models based on different understandings or beliefs about how the system works. The principal goal of this approach is to understand how the structure of a system (ie the variables and the signs of their connecting links) affects its dynamics. This is achieved through analysis of a system's feedback properties in predicting how it will respond to a perturbation.

Statistical models emphasise generality and precision, they are more easily tested and will provide thresholds with estimates of uncertainty. These are critical to setting quantitative thresholds and trigger points and providing the analysis needed to refine ecosystem level analyses. However, they have difficulty in describing the complexity of ecosystems, and more particularly, cannot address questions of causation. Statistical models have been used to address questions around single sector activities and outcomes – such as fisheries impacts (eg Trenkel and Rochet 2010, Rochet et al. 2010, Foster et al. 2014) and acoustic impacts (Pine et al. 2014). They are most useful when there is a direct measure of the value of interest and monitoring data can be collected quickly and cheaply and the response of the system to the pressures is sufficient to clearly detect the signal of change.

In contrast, numerical simulation models are able to capture a significant amount of ecological complexity of the systems and can also incorporate the dynamics of human activities, they are, however, less easily tested and may require significantly more data. Initially these models were focused primarily on fisheries and their trophic interactions with other biological elements of marine ecosystems, though some of the more sophisticated represented the gross pressure of other activities (such as coastal development and catchment based nutrient flows) as background to the fisheries work (Fulton 2011, Fulton et al 2014). A diversity of approaches have been applied, from multispecies models with environmental and social drivers to full end-to-end (or whole of system) ecosystem models that include the physical environment, habitats, food webs and all the human uses (Little et al 2006, Fulton et al 2011, Plagányi et al, 2011, 2014). Simulation models require significant amounts of data, for all parts of the model, to support the specification of parameters and to support assumptions about the functional forms of ecological relationships. They have an advantage that they can portray the ecosystem in a way that resonates with stakeholders. This in itself can lead to an improved and shared understanding that can remove disagreements on potential management actions. However, while representation of the uncertainty around simulation model results is improving, unless large scale ensemble-modelling approaches are used, it is still difficult to determine the confidence in the models in terms of structural and parametric uncertainty. Moreover, simulation models can be good at describing the current state of the ecosystem, but may have limited skill in distinguishing the relative probability of future states. One approach showing significant potential is the 'minimum realistic' or intermediate complexity approach (MICE, Plagányi et al 2014). By focusing only on the relevant subsystem proven model fitting methods can be used and skill assessments can be undertaken, providing greater confidence in model results for some loss of generality.

Table 4: Examples of Level 3

Examples of Level 3	Description	Reference
Sustainability and impacts of fisheries.	Statistical models are used to address changes in species abundances and stock levels, at either single or multispecies levels. These models provide strong tactical advice on management over short to moderate time frames but only weakly capture ecosystem level properties.	Foster et al. (2014), Trenkel and Rochet (2010), Rochet et al. (2010)
Ecosystem modelling of alternative management options of Australia's Southern and Eastern Scalefish and Shark Fishery	The Atlantis modelling framework has been used in southeastern Australia to look at alternative management options for the major fisheries there. The first round of simulations provided informed a management restructure for the region in 2005 and more recent studies have identified how cumulative pressures from climate change, coastal development and multiple sector activities in the region could undermine future system state under a range of management strategies.	Fulton et al. 2014, Fulton and Gorton 2014
Models of Intermediate Complexity for Ecosystem assessments' (MICE)	MICE models combine an understanding of a limited subset of an ecosystem with statistical approaches to model fitting. The models have a tactical focus and attempt to capture stakeholders understanding of how the system operates.	Plagányi et al. (2014)

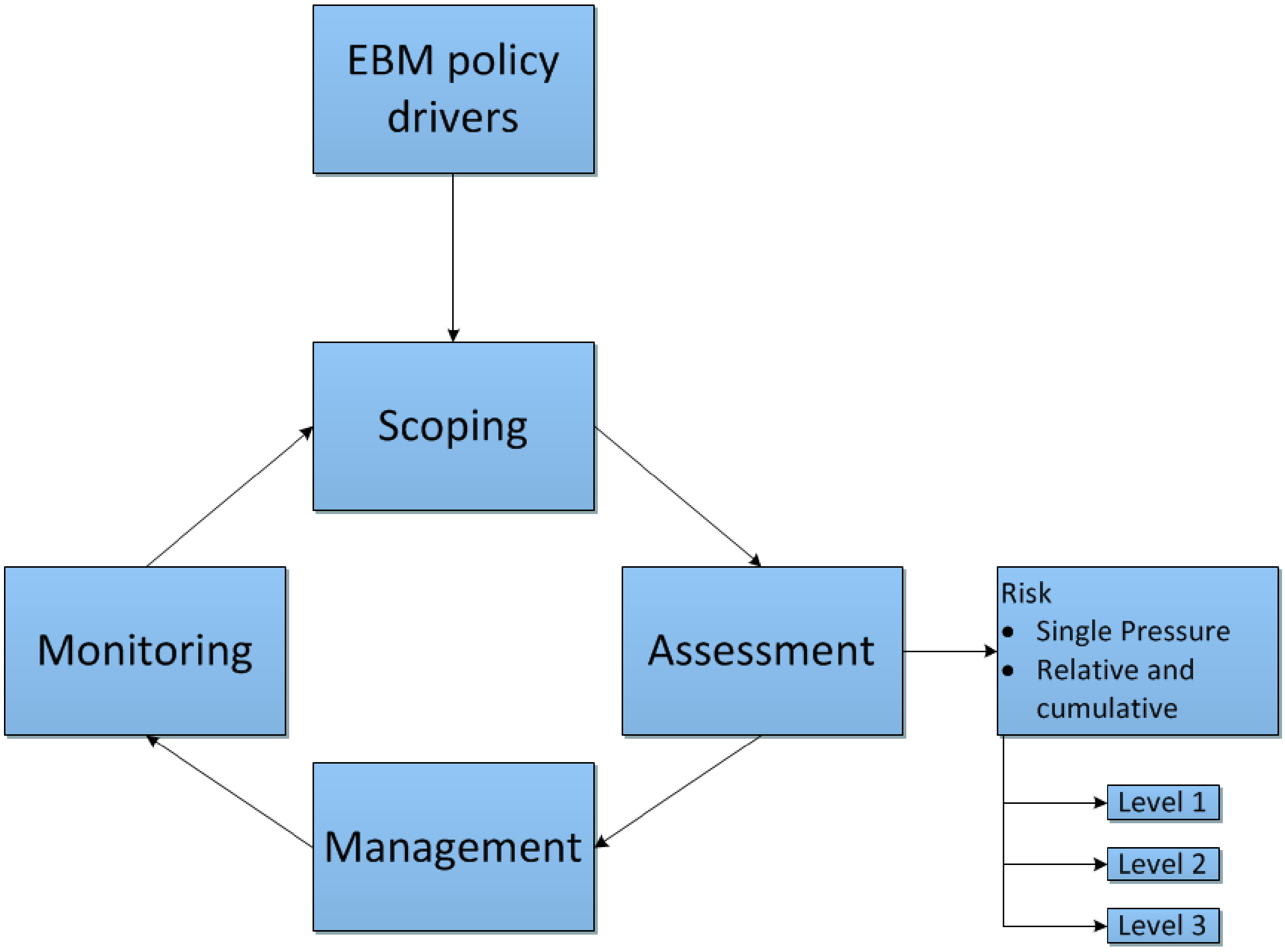


Figure 2: A simplified framework for adaptive management. Within this loop the key points for the insertion of scientific knowledge is in assessment and monitoring

5. DISCUSSION

A structured process for risk assessment is a key component of any adaptive EBM cycle (Figure 2). The different definitions and measures of risk sometimes reflect different lexicons but more often reflect fundamental methodological differences driven by facets such as the objectives and scope of the study, and the availability of data. When there are impacts from a single sector or on a single species, with no secondary impacts (ie by-catch, a key predator or biogenic habitat), it will be not always be necessary to consider the full ecosystem. However, where there are multiple sectors, impacting multiple species, then an EBM risk based approach is appropriate. A hierarchical structured risk assessment has previously been primarily used within fisheries domains (eg Hobday et al. 2011, Fletcher and Bianchiia 2014, Levin et al. 2009), but should be equally applicable to other management issues within the marine domain. The flexibility provided by such an approach means that the risk assessment process can take many forms, depending on the issues being considered risk

Any risk assessment process should sit within a broader framework of adaptive EBM (eg Anthony et al. 2013, Fletcher and Bianchi 2014). The two key points for the insertion of science into an adaptive EBM cycle are into risk assessments and into monitoring of ecosystem responses to management. As illustrated in this paper, scientific input into risk assessment is critical to the effective functioning of an adaptive EBM cycle (figure 2). Within the risk assessment phase the steps identified in this paper can be carried out to effectively inject current or new scientific understanding into the decision process to support or reduce ambiguity around decisions and manage expectations around how and when additional science is needed for more complex decisions. In the same way, there are multiple steps within the monitoring phase of the EBM cycle (Hayes et al. 2015). When the monitoring and assessment phases are linked through the management phase, with the identification of appropriate objectives and triggers, the adaptive EBM cycle can be completed.

The hierarchical framework proposed in this paper provides scientists and policy makers with guidance and a common lexicon for assessing cumulative risks and estimating impacts to marine ecosystems. The framework provides for a cost-effective and consistent approach to assessments, accommodating a broad range of marine environment assessment cases, leading to priorities for action. The approach acknowledges the importance of ecosystem models for estimating cumulative risks and provides a frame for understanding how they can be cost-effectively and consistently applied to estimate impacts and improve understanding.

6. REFERENCES

Anthony, KRN, Dambacher, JM, Walshe, T, Beeden, R. (2014). A Framework for Understanding Cumulative Impacts, Supporting Environmental Decisions and Informing Resilience---Based Management of the Great Barrier Reef World Heritage Area. Final Report to the Great Barrier Reef Marine Park Authority and Department of the Environment.

Astles, KL, Holloway, MG, Steffe, A, Green, M, Ganassin, C, and Gibbs, PJ. (2006). An ecological method for qualitative risk assessment and its use in the management of fisheries in New South Wales, *Australia*, Fisheries Research 82 : 290-303.

Bax, N, Williams, A, Davenport, S, Bulman, C. (1999). Managing the ecosystem by leverage points: a model for a multispecies fishery. Ecosystem Approaches for Fisheries Management. 1:283-303

CBD (2008). DECISION ADOPTED BY THE CONFERENCE OF THE PARTIES TO THE CONVENTION ON BIOLOGICAL DIVERSITY at its ninth meeting. Decision IX/20, 2008.

Coll, M, Piroddi, C, Albouy, C, Ben Rais Lasram, F, Cheung, WWL, Christensen, V, Karpouzi, VS, Guilhaumon, F, Mouillot, D, Paleczny, M, Palomares, ML, Steenbeek, J, Trujillo, P, Watson, R, and Pauly, D. (2012). The Mediterranean Sea under siege: spatial overlap between marine biodiversity, cumulative threats and marine reserves. Global Ecology and Biogeography 21: 465-480.

Dambacher, J, Hayes, KR, Hosack, GR, Lyne, V, Clifford, D, Dutra, LXC, Moeseneder, CH, Palmer, MJ, Sharples, R, Rochester, WA, Taranto, TJ, and Smith, R. (2012). Project Summary: National Marine Ecological Indicators.

Dambacher, J, Young, J, Olsen, R, Allain, V, Galvan-Magana, F, Lansdel, M, Bocanegra-Castillo, N, Alatorre-Ramirez, V, Cooper, S, and Duffy, L. (2010). Analyzing pelagic food webs leading to top predators in the Pacific Ocean: A graph-theoretic approach. Progress in Oceanography 86: 152-165.

Dambacher, J M, Gaughan, D J, Rochet, M J, Rossignol, PA, and Trenkel, VM. (2009). Qualitative modelling and indicators of exploited ecosystems, Fish and Fisheries 10 : 305-322.

Dambacher, JM, Rothlisberg, PC, and Loneragan, NR. (2015). Qualitative mathematical models to support ecosystem-based management of Australia's Northern Prawn Fishery. Ecological Applications 25:278–298. http://dx.doi.org/10.1890/13-2030.1

Department of Sustainability, Environment, Water, Population and Communities – DSEWPaC. (2012). Marine bioregional plan for the South-west Marine Region. http://www.environment.gov.au/marine/marine-bioregional-plans

Eken, G, Bennun, L, Brooks, T, Darwall, W, Fishpool, L, Foster, M, Knox, D, Langhammer, P, Matiku, P, Radford, E, Salaman, P, Sechrest, W, Smith, M, Spector, S, and Tordoff, A. (2004). Key Biodiversity Areas as Site Conservation Targets. BioScience 54 : 1110-1118.

Fletcher, W, and Bianchi, G. (2014). The FAO e EAF toolbox: Making the ecosystem approach accessible to all fisheries. Ocean & Coastal Management 90: 20-26.

Fletcher, W J. (2005). The application of qualitative risk assessment methodology to prioritise issues for fisheries management, ICES Journal of Marine Science 62: 1576-1587.

Foster, SD, Dunstan PK, Althaus F and Williams, A. (2014). The cumulative effect of trawl fishing on a multispecies fish assemblage in south-eastern Australia. Journal of Applied Ecology 52: 129-139. doi: 10.1111/1365-2664.12353

Fulton, EA, Link, JS, Kaplan, IC, Savina-Rolland, M, Johnson, P, Ainsworth, C, Horne, P, Gorton, R, Gamble, RJ, Smith, ADM and Smith, DC. (2011). Lessons in modelling and management of marine ecosystems: the Atlantis experience. Fish and Fisheries, 12: 171–188. doi: 10.1111/j.1467-2979.2011.00412.x

Fulton, EA, Smith ADM, Smith, DC and Johnson, P. (2014). An Integrated Approach Is Needed for Ecosystem Based Fisheries Management: Insights from Ecosystem-level Management Strategy Evaluation. PLoS ONE 9(1): e84242. doi:10.1371/journal.pone.0084242

Fulton, EA and Gorton, R. (2014). Adaptive Futures for SE Australian Fisheries & Aquaculture: Climate Adaptation Simulations. CSIRO, Australia.

Garthwaite, PH and O'Hagan, A. (2000). Quantifying expert opinion in the UK water industry: an experimental study. The Statistician 49(4): 455-477

Garthwaite, PH, Kadane, JB and O'Hagan, A. (2005). Statistical Methods for Eliciting Probability Distributions. Journal of the American Statistical Association 100:470, 680-701, DOI:10.1198/016214505000000105

Halpern, BS, Kappel, CV, Selkoe, KA, Micheli, F, Ebert, CM, Kontgis, C, Crain, CM, Martone, RG, Shearer, C, and Teck, SJ. (2009). Mapping cumulative human impacts to California Current marine ecosystems. Conservation Letters 2 : 138-148.

Halpern, BS, Walbridge, S, Selkoe, KA, Kappel, CV, Micheli, F, D'Agrosa, C, Bruno, JF, Casey, KS, Ebert, C, Fox, HE, Fujita, R, Heinemann, D, Lenihan, HS, Madin, EMP, Perry, MT, Selig, ER, Spalding, M, Steneck, R and Watson, R. (2008). A global map of human impact on marine ecosystems. Science 319: 948-952.

Hayes, KR, Clifford, D, Moeseneder, C, Palmer, M, and Taranto, T. (2012). National Indicators of Marine Ecosystem Health: Mapping Project. A report prepared for the Australian Government Department of Sustainability, Environment, Water, Population and Communities. CSIRO Wealth from Oceans Flagship, Hobart.

Hayes, KR, Dambacher, JM, Hosack, GR, Bax, NJ, Dunstan, PK, Fulton, E, Thompson, PA, Hartog, JR, Hobday, AJ, Bradford, R, Foster, SD, Hedge, P, Smith, DC, Marshall, CM. (2015). Identifying indicators and essential variables for marine ecosystems. Ecological Indicators 57: 409–419

Hilborn, R, Maguire, JJ, Parma, AM, and Rosenberg, A A. (2001). The Precautionary Approach and risk management: can they increase the probability of successes in fishery management? Canadian Journal of Fisheries and Aquatic Science 58 : 99-107.

Hobday, AJ, Smith, A, Stobutzki, I, Bulman, C, Daley, R, Dambacher, J, Deng, R, Dowdney, J, Fuller, M, Furlani, D, Griffiths, S, Johnson, D, Kenyon, R, Knuckey, I, Ling, S, Pitcher, R, Sainsbury, K, Sporcic, M, Smith, T, Turnbull, C, Walker, T, Wayte, S, Webb, H, Williams, A, Wise, B, and Zhou, S. (2011). Ecological risk assessment for the effects of fishing. Fisheries Research 108 : 372-384.

Hosack, G. and Dambacher, J. (2012). Ecological Indicators for the Exclusive Economic Zone of Australia's South East Marine Region. A report prepared for the Australian Government Department of Sustainability, Environment, Water, Population and Communities. CSIRO Wealth from Oceans Flagship.

Kadane, JB, and Wolfson LJ. (1998). Experiences in Elicitation. The Statistician 47(1): 3-19

Kynn M. (2008). The 'heuristics and biases' bias in expert elicitation. Journal of the Royal Statistical Society A 171(1): 239–264

Levin PS, Fogarty MJ, Murawski SA, Fluharty D. (2009). Integrated ecosystem assessments: Developing the scientific basis for ecosystem-based management of the ocean. PLoS Biol 7(1): e1000014. doi:10.1371/journal.pbio.1000014

Levins, R. (1966). The Strategy of Model Building in Population Biology, American Scientist 54 : 421-431.

Levins, R. (1974). The qualitative analysis of partially specified systems, Annals of the New York Academy of Sciences 231: 123–138.

Levins, R. (1998). Qualitative mathematics for understanding, prediction, and intervention in complex systems. In: Rapport, D., Costanza, R., Epstein, P.R., Gaudet, C., Levins, R. (Eds.), Ecosystem Health. Blackwell Science, Malden, Massachusetts, USA, pp. 178–204

Little, LR, Fulton, EA, Gray, R, Hayes, D, Lyne, V, Scott, R, Sainsbury, SK, and McDonald, AD. (2006). Multiple Use Management Strategy Evaluation for the North West Shelf: Results and Discussion. North West Shelf Joint Environmental Management Study Technical Report, vol. 18.

Ottersen, G, Olsen, E, van de Meeren, GI, Dommasnes, A and Loeng, H. (2011). The Norwegian plan for integrated ecosystem-based management of the marine environment in the Norwegian Sea. Marine Policy 35: 389–398

Pine, MK, Jeffs, AG, and Radford, CA. (2014). The cumulative effect on sound levels from multiple underwater anthropogenic sound sources in shallow coastal waters. Journal of Applied Ecology 2014, 51, 23–30 doi: 10.1111/1365-2664.12196

Plagányi, ÉE, Bell, JD, Bustamante, RH, Dambacher, JM, Dennis, DM, Dichmont, CM, Dutra, LXC, Fulton, EA, Hobday, AJ, van Putten, IE, Smith, F, Smith, ADM and Zhou, S (2011). Modelling climate change effects on Australian and Pacific aquatic ecosystems: a review of analytical tools and management implications. Marine and Freshwater Research, 62: 1132-1147.

Plagányi, É, Punt, A, Hillary, R, Morello, E, Thébaud, O, Hutton, T, Pillans, R, Thorson, J, Fulton, EA, Smith, ADM, Smith, F, Bayliss, P, Haywood, M, Lyne, V and Rothlisberg, P. (2014). Multi-species fisheries management and conservation: tactical applications using models of intermediate complexity. Fish and Fisheries 15:1-22. DOI:10.1111/j.1467-2979.2012.00488.x

Puccia CJ and Levins, R. (1985). Qualitative modeling of complex systems: an introduction to loop analysis and time averaging. Harvard University Press, Cambridge, USA.

Rochet, MJ, Trenkel, VM, Carpentier, A, Coppin, F, De Sola, LG, Léauté, JP, Mahé, JC, Maiorano, P, Mannini, A, Murenu, M, Piet, G, Politou, CY, Reale, B, Spedicato, MT, Tserpes, G and Bertrand, JA. (2010). Do changes in environmental and fishing pressures impact marine communities? An empirical assessment. Journal of Applied Ecology, 47: 741–750. doi: 10.1111/j.1365-2664.2010.01841.x

Scandol, J, Ives, M, and Lockett, M. (2009). Development of national guidelines to improve the application of risk-based methods in the scope, implementation and interpretation of stock assessments for data-poor species. FRDC Project 2007/016. Industry & Investment NSW –Fisheries Final Report Series 115, 1884pp.

Sethi, S. (2011). Risk management for fisheries, Fish and Fisheries 11 : 341-365.

Sharp, BR, Parker, SJ, and Smith, N. (2009). An impact assessment framework for bottom fishing methods in the CMALR convention area, CCAMLR Science 16 : 195-210.

Secretariat of the Pacific Community. (2010). A community-based ecosystem approach to fisheries management: guidelines for Pacific Island Countries. Compiled by the Secretariat of the Pacific Community. www.spc.int/DigitalLibrary/Doc/FAME/Manuals/Anon_10_EAFguidelines.pdf

Smith, ADM, Fulton, EJ, Hobday, AJ, Smith, DC, and Shoulder, P. (2007). Scientific tools to support the practical implementation of ecosystem-based fisheries management. ICES Journal of Marine Science, 64: 633–639.

Smith, ADM, Smith, DC, Tuck, GN, Klaer, N, Punt, AE, Knuckey, I, Prince, J, Morison, A, Kloser, R, Haddon, M, Wayte, S, Day, J, Fay, G, Pribac, F, Fuller, M, Taylor, B, Little, LR. (2008). Experience in implementing harvest strategies in Australia's south-eastern fisheries, Fisheries Research, 94(3):373-379. http://dx.doi.org/10.1016/j.fishres.2008.06.006.

Smith, ADM, Smith DC, Haddon, M, Knuckey, I, Sainsbury, KJ, and Sloan, S. (2014). Implementing harvest strategies in Australia: 5 years on. ICES Journal of Marine Science, 71: 195–203.

Trenkel, VM, and Rochet, MJ. (2010). Combining time trends in multiple metrics for identifying persistent changes in population processes or environmental stressors. Journal of Applied Ecology 47:751–758

National Environmental Research Program

MARINE BIODIVERSITY hub

www.nerpmarine.edu.au

Contact: Piers Dunstan
CSIRO Oceans and Atmosphere
Castray Esplanade, Hobart
piers.dunstan@csiro.au
+61 3 6232 5382